

III. SCIENTIFIC BACKGROUND

A. PURPOSE

The purpose of this section is to provide general scientific background information needed to understand some of the material presented and discussed elsewhere in this report. Information is provided for air pollutants, deposition of air pollutants from the atmosphere to terrestrial and aquatic ecosystems, aquatic receptors, terrestrial receptors that are sensitive to air pollutant effects, and finally visibility. Each topic is discussed in general terms. Shenandoah National Park (SHEN)-specific information is presented in subsequent sections of the report.

B. GASEOUS POLLUTANTS

1. Ground-level Ozone

Ground-level (tropospheric) ozone (O_3) is a particular concern for national park lands, including SHEN, because remote forested areas can experience greater cumulative O_3 exposure, higher minimum values, and higher maximum concentrations than the upwind urban and industrial areas where most of the pollutant sources are located. In many places, O_3 concentrations tend to increase with elevation (Brace and Peterson 1998, Cooper and Peterson 2000), although there are some exceptions (Bytnerowicz et al. 1999). This can lead to elevated values in mountainous locations (Aneja et al. 1991).

Ground-level ozone is a secondary pollutant formed by the photo-oxidation of nitrogen oxides (NO_x) and volatile organic compounds (VOCs). Ozone is a colorless gas and is a component of smog, which can develop during the clear warm weather associated with high pressure systems. It is an important regional pollutant because it forms during transport of its precursors and can occur at high concentrations in areas remote from precursor sources. The level of O_3 in a “pristine” area may have a weekly average as low as 10 to 25 parts per billion by volume (ppbv; Altshuller and Lefohn 1996, Cooper and Peterson 2000), with maximum hourly O_3 concentrations generally less than about 50 to 80 ppbv. Ozone is injurious to human health as well as to vegetation. It is an eye, nose, and lung irritant and causes damage to plants. Ozone is a potential threat to high-elevation plant species because concentrations and total exposure can be higher at higher elevations under appropriate atmospheric conditions (Loibl et al. 1994; Sandroni et al. 1994; Brace and Peterson 1996, 1998). Dispersion and transport of pollutants vary locally, but there are clearly periods of high O_3 exposure in and around SHEN every

summer. The challenge is to quantify the spatial distribution of this exposure relative to the location of sensitive receptors.

In order to protect human health and welfare, the U. S. EPA has established primary and secondary O₃ National Ambient Air Quality Standards for maximum allowable O₃ concentration levels. Prior to 1997, these standards were based upon 1-hour average O₃ measurements. They were revised in 1997, when the EPA promulgated new standards, both primary and secondary, based upon an 8-hour average value. Under this rule, the annual fourth-highest daily maximum 8-hour O₃ concentration, averaged over three years, must not exceed 0.08 ppm (parts per million; 80 ppbv, which can be rounded up to 85 ppbv). This average is computed by first determining the highest 8-hour average O₃ value for each day of the year, and then identifying the 4th-highest of all daily maximum 8-hour O₃ values that occurred during the year. These 4th-highest values are then averaged over three successive years to determine the final concentration value that is compared to the standard. Because of rounding procedures, EPA has indicated that 85 ppbv is the lowest concentration that would exceed the 80 ppbv standard. The new secondary standard is identical to the primary standard.

The decision by the U.S. EPA to transition from an O₃ standard based upon a 1-hour average to a standard based upon an 8-hour average was prompted by research indicating that prolonged exposure to O₃ at concentrations lower than the 1-hour standard can have significant impacts upon human health. There is also research indicating that prolonged exposure to O₃ is particularly harmful to vegetation. The EPA has recognized the potential usefulness of some type of exposure index in setting the secondary O₃ standard to protect agricultural crops and native vegetation (U.S. EPA 1997). In its final ruling on the new O₃ standard, the EPA recognized that a cumulative index measuring long-term O₃ exposure is very likely to be more relevant to vegetation than the 8-hour standard that was chosen, but concluded that there were not yet enough data to specify a new secondary standard based upon such an index. It did, however, indicate that it would focus additional research on this issue and reevaluate the need for a seasonal cumulative index in the next review of the O₃ standard.

Two such indices of O₃ exposure are presented here. The first is the SUM06. This index is computed by adding all hourly O₃ concentration values greater than or equal to 0.06 ppm. Because plants are most active photosynthetically during daylight hours in the summer time, and thus most likely to take in O₃ and other gases during these hours, the index was calculated here using values measured during the 12 hours from 8 AM to 8 PM in the months of May through

September of each year. A similar index is the W126 index. This index is the sum of all hourly O₃ concentrations where each concentration is weighted by a function that gives greater emphasis to the higher hourly concentrations while still including the lower ones (Appendix A).

2. Sulfur Dioxide

Sulfur dioxide (SO₂) is a product of fossil fuel, primarily coal and oil, combustion. Some of the largest emitters are coal-fired electric power plants and smelters. Although more stringent regulations have reduced emissions over the last 30 years, SO₂ continues to be a pollutant of concern in many areas of the United States, including throughout the Southeast. SO₂ is a precursor of pollutants which cause acidic deposition and visibility degradation.

Like O₃, SO₂ is a gas and enters the leaf through the stomata. Inside the leaf it disrupts mesophyll cell functioning, causing reduced productivity of the leaf. SO₂ injury in plants is characterized by leaf bleaching and chlorosis, necrotic lesions, and early senescence. Prolonged exposure can weaken a plant, making it susceptible to pathogens and other organisms. Some species are sensitive to chronic exposures as low as 25 ppbv (Treshow and Anderson 1989).

There are few data on the effects of sulfur (S) compounds on mature trees or other native plants, and there is a wide range of sensitivities to ambient S compounds (Davis and Wilhour 1976, Smith 1990). Limited data on tree seedlings (Hogsett et al. 1985) suggest that SO₂ concentrations below 20 ppbv (24-hour mean) do not produce visible injury symptoms (Peterson and Sullivan 1998). In addition to potential direct effects on vegetation, atmospheric S compounds contribute to visibility degradation and to S deposition that has been associated with effects on aquatic ecosystems.

3. Volatile Organic Compounds and Nitrogen Oxides

Volatile organic compounds and NO_x are important precursors of O₃. Several VOCs fall under the Hazardous Air Pollutant section of the Clean Air Act and VOCs must be addressed as precursors to O₃ under State Implementation Plans. NO_x is also a precursor for pollutants that cause acidic deposition and visibility degradation. VOCs can also contribute to visibility degradation.

All high-temperature combustion processes emit NO_x. Automobiles and stationary fossil fuel burning systems are the major anthropogenic (human-caused) sources of NO_x emissions in

the United States. Naturally-occurring NO_x compounds originate from soils, wildfire, lightning and decomposition.

Anthropogenic sources of VOCs include motor vehicle exhaust, gasoline vapors, stationary fuel combustion, commercial and industrial processes, and emissions from solid wastes (Smith 1990). Natural systems, particularly soils and vegetation, produce VOCs and emit them to the atmosphere; trees in particular emit the highly reactive hydrocarbons isoprene and terpene. Globally, biogenic sources of VOCs exceed anthropogenic sources, although anthropogenic sources typically dominate in localized, urban areas. VOCs include a large number of hydrocarbons which vary greatly in reactivity.

Ozone is formed from the reaction of NO_x in the presence of sunlight and VOCs. In unpolluted air, reactions involving NO, nitrogen dioxide (NO₂), and O₃ form a chemical cycle that does not lead to net loss or production of O₃. VOCs, both human-made and natural, interfere with this cycle, allowing for the buildup of O₃. Emitted NO rapidly converts to NO₂ in the atmosphere. The latter is a criteria pollutant. Both NO_x (the sum of NO and NO₂) and VOCs are thus precursors to O₃ formation. This buildup proceeds during the daytime when ultraviolet radiation from the sun is available to drive the required reactions, and ceases at night. When fresh NO is available at night, it reacts with O₃ and removes it from the atmosphere. This creates a pronounced diurnal cycle in O₃ concentrations in many areas.

C. ATMOSPHERIC DEPOSITION

Given the structural complexity of landforms and the broad distribution of major pollution source areas, a network of air quality and atmospheric deposition monitors in or near Class I areas is needed to fully evaluate the spatial patterns of air quality and atmospheric deposition within SHEN, or any other Class I area. Most existing air quality and deposition data are short-term and/or are from urban areas remote from wildland locations, although general patterns of nitrogen (N) and S deposition can be inferred from state and national databases. It is known that air pollution and deposition in some areas have increased considerably during the last 30 to 40 years, while decreasing in others.

The estimation of deposition of atmospheric pollutants in remote areas is especially difficult because all components of the deposition (rain, snow, cloudwater, dryfall and gases) have seldom been measured concurrently. Even measurement of wet deposition remains a problem in some areas because of the logistical difficulties in operating a site at remote locations. Portions

of the wetfall have been measured by using snow cores (or snow pits), bulk deposition (open containers), and automated sampling devices such as those used at the National Atmospheric Deposition Program/National Trends Network (NADP/NTN) sites. All of these approaches suffer from limitations that cause problems with respect to developing annual deposition estimates. The snow sampling includes results for only a portion of the year and may seriously underestimate the load for that period if there is a major rain-on-snow event. Bulk deposition samplers are subject to contamination problems from birds and litterfall. Weekly data on dry deposition are available at some sites (including Big Meadows within SHEN), collected within the Clean Air Status and Trends Network (CASTNet). Dryfall from wind-borne soil can constitute major input to the annual deposition load, particularly in arid environments. Aeolian inputs from dryfall can provide a major source of acid neutralization not generally measured in other forms of deposition. Gaseous deposition is calculated from the product of ambient air concentrations and estimated deposition velocities. However, the derivation of deposition velocities is subject to considerable debate.

The need to measure or estimate different forms of atmospheric deposition complicates the difficult task of measuring and monitoring total deposition. Cloudwater can be an important portion of the hydrologic budget in coastal areas and some high-elevation forests (Harr 1982, Lovett and Kinsman 1990), and failure to capture this portion of the deposition input could lead to underestimation of annual deposition. Furthermore, cloudwater chemistry has the potential to be more acidic than rainfall (Weathers et al. 1988). Due to these limitations, there is considerable uncertainty regarding the amount of current deposition (wet, dry, cloud) of atmospheric pollutants in any of the Class I areas in the United States.

Dry deposition approximates, or is slightly less than, wet deposition for most ions at most sites; for example, dry deposition constituted 30 to 50% of total deposition of most ions at most Integrated Forest Study (IFS) sites throughout the eastern United States (Johnson and Lindberg 1992). Cloud water deposition contributed substantially to total ionic deposition at only two high-elevation IFS sites: Whiteface Mt., NY (1,000 m) and near Clingman's Dome, NC (1,740 m), ranging from about 20% of total for base cations at Whiteface Mt. to 50% of total for sulfate (SO_4^{2-}) at the North Carolina site. Estimates of wet, dry, and cloud deposition were generated by Shannon (1998) at multiple sites throughout the southeastern United States using the Advanced Statistical Trajectory Regional Air Pollution (ASTRAP) model, suggesting that the sum of dry plus cloud deposition was approximately equal to wet deposition, for both S and N, throughout

the region (Sullivan et al. 2002a). These model estimates are in general agreement with data derived from NADP and CASTNet monitoring at Big Meadows in SHEN (Section V).

D. AQUATIC RESOURCES AND SENSITIVE INDICATORS

1. Water Chemistry

Water bodies are regarded by residents of the southern Appalachian Mountains region as extremely important (SAMAB 1996), and multiple uses include drinking water, fishing, other aquatic recreation, transportation, livestock watering, irrigation, flood control, hydroelectric power, wildlife observation, and waterfront human habitation. There is general agreement that water quality has improved in the southern Appalachians since the adoption of the Clean Water Act in 1972. However, the rate of improvement has recently slowed since most municipal and industrial discharges now control point source pollution, and the remaining nonpoint sources (storm water runoff, sediment contamination, acid deposition, and spills) are more difficult and expensive to control. Future water quality in some areas is likely to be challenged by population growth (Wallace et al. 1992, SAMAB 1996).

Stream concentrations of major ions depend on atmospheric inputs, soil pools of elements, in-stream processing, and weathering rates of rocks, including mechanical breakage of rocks and chemical processes that alter minerals by adding or removing elements. Chemical weathering is more important than mechanical weathering as a source of base cations for the soil cation exchange pool because chemical processes are more effective on smaller particles, which have greater overall surface area than do larger particles (Isphording and Fitzpatrick 1992).

The potential effects of S deposition on surface water quality have been well studied throughout the United States, particularly within the EPA's Aquatic Effects Research Program, a component of the National Acid Precipitation Assessment Program (NAPAP). Major findings were summarized in a series of State of Science and Technology Reports (e.g., Baker et al. 1990a,b; Sullivan 1990; Turner et al. 1990) and the final NAPAP policy report, the Integrated Assessment (NAPAP 1991). More recent findings were summarized in Church and Van Sickle (1999) and in the review of Sullivan (2000), which provides the basis for much of the material presented in the remainder of this section. Although aquatic effects from N deposition have not been studied as thoroughly as those from S deposition, concern has been expressed regarding the role of nitrate (NO_3^-) in acidification of surface waters (particularly during hydrologic episodes), the role of NO_3^- in the long-term acidification process, the contribution of ammonium (NH_4^+)

from agricultural sources to surface water acidification, and the potential for anthropogenic N deposition to stimulate eutrophication of freshwaters and estuaries (e.g., Sullivan 1993, 2000; Wigington et al. 1993; Sullivan et al. 1997; NAPAP 1998)

Computer models can be used to predict pollution effects on aquatic ecosystems and to perform simulations of future ecosystem response (c.f., Cosby et al. 1985a,b,c). The Model of Acidification of Groundwater in Catchments (MAGIC), a lumped-parameter mechanistic model, has been used throughout North America and Europe and extensively tested against the results of diatom reconstructions of historic water chemistry and ecosystem manipulation experiments (e.g., Wright et al. 1986; Sullivan et al. 1992, 1996; Sullivan and Cosby 1995; Cosby et al. 1995, 1996; Sullivan 2000). MAGIC provided the basis for aquatic effects modeling for this assessment, although other models are also available (c.f., Christophersen et al. 1982; Liu et al. 1991, 1992; Aber et al. 1997). Watershed models are valuable to management agencies which require quantitative predictions of air pollutant impacts and the likely future benefits of emissions control programs. Nitrogen dynamics have more recently been added to the MAGIC model (Jenkins et al. 1997).

Atmospheric deposition of S and N (as NO_3^- and as NH_4^+ , which can be quickly nitrified to NO_3^-) often cause increased concentrations of SO_4^{2-} in drainage waters and can, in some cases, cause increased concentrations of NO_3^- . An increase in the concentration of either of these mineral acid anions will generally result in a number of additional changes in water chemistry. These can include:

- increased concentration of base cations (Ca^{2+} , Mg^{2+} , K^+ , Na^+)
- decreased acid neutralizing capacity (ANC)
- increased concentration of hydrogen ion (H^+ ; decreased pH)
- increased concentration of dissolved aluminum (Al)

Increased leaching of base cations along with SO_4^{2-} and/or NO_3^- can lead to base cation depletion in base-poor watershed soils. In such cases, base cation concentrations in drainage waters will eventually decrease, sometimes to levels below those found prior to the initiation of acidic deposition. Increased concentrations of H^+ and/or Al in surface water occur only in response to higher concentrations of SO_4^{2-} or NO_3^- when ANC has decreased to less than about 50 to 100 $\mu\text{eq/L}$. At higher ANC values, contributions of SO_4^{2-} or NO_3^- are mainly balanced by increasing base cation concentrations in runoff and some decrease in ANC. High concentrations of H^+ or Al can be toxic to fish and other aquatic biota.

If NO_3^- leaches into stream or lakewater as a result of increased N deposition, the result can be eutrophication or acidification. If N is limiting for aquatic primary production, the added NO_3^- can result in increased algal productivity, which can cause disruption of aquatic community dynamics. If N is not limiting, then the added NO_3^- will remain in solution. If the added NO_3^- is not balanced by an increase in base cation concentration, acidification will occur.

The Southeast was not glaciated during the last glacial period (about 100,000 to 10,000 years ago). As a result, the soils of the Southeast are largely residual, relatively deep, and highly structured vertically, versus those in glaciated areas. The lower horizons of southeastern soils are also rich in iron and Al, which can strongly affect stream chemistry via efficient retention of many negatively charged solutes, of which SO_4^{2-} is of major interest in the context of acidification. Ultisols represent one of the dominant soil groups in the Southeast; these are characterized by surface and subsurface horizons that are loamy or clayey in texture. They are typically acidic soils low in base saturation (Adams and Hackney 1992).

Although NO_3^- is also associated with precipitation acidity, NO_3^- concentrations in upland surface waters are commonly low due to demand for N as a nutrient, especially in regenerating forests (Aber et al. 1989, 1998; Johnson and Lindberg 1992). The deposition and fate of S in watersheds has accordingly received most of the attention in regional-scale assessments of acidic-deposition effects on aquatic ecosystems (e.g., NAPAP 1991, Church et al. 1992, Sullivan et al. 2002a).

Surface waters that are sensitive to acidification from acidic deposition of S or N typically exhibit a number of characteristics. Such characteristics either predispose the waters to acidification and/or correlate with other parameters that predispose the waters to acidification. Although precise guidelines are not widely accepted, general ranges of parameter values that reflect sensitivity are as follows (c.f., Sullivan 2000):

Dilute - Waters have low concentrations of all major ions, and therefore specific conductance is low ($< 25 \mu\text{S}/\text{cm}$). Dilute waters are low in base cation concentrations, suggesting small cation pools in watershed soils.

Acid neutralizing capacity - ANC is low. Acidification sensitivity has long been defined as $\text{ANC} < 200 \mu\text{eq}/\text{L}$, although more recent research has shown this criterion to be too inclusive. Waters sensitive to chronic acidification generally have $\text{ANC} < 50 \mu\text{eq}/\text{L}$, and waters sensitive to episodic acidification generally have $\text{ANC} < 100 \mu\text{eq}/\text{L}$.

Base cations - Concentrations are low in non-acidified waters, but can increase substantially in response to acidic deposition. In relatively pristine areas, the concentration of base

cations ($\text{Ca}^{2+} + \text{Mg}^{2+} + \text{Na}^+ + \text{K}^+$) in sensitive waters will generally be less than about 50 to 100 $\mu\text{eq/L}$.

Organic acids - Concentrations are low in waters sensitive to the effects of acidic deposition. Dissolved organic carbon (DOC) and associated organic acids cause water to be naturally low in pH and ANC, or even to be acidic ($\text{ANC} < 0$), but also impart substantial pH buffering at these low pH values.

pH - pH is low, generally less than 6.0 to 6.5 in acid-sensitive waters. In areas that have received substantial acidic deposition, paleolimnological studies have shown that acidified lakes are generally those that had pre-industrial pH between 5 and 6.

Acid anions - Sensitive waters generally do not have large contributions of mineral acid anions (e.g., SO_4^{2-} , F^- , Cl^-) from geological or geothermal sources. In particular, the concentration of SO_4^{2-} in drainage waters would not be substantially higher than could reasonably be attributed to atmospheric inputs, after accounting for probable dry deposition and evapotranspiration.

Physical characteristics - Sensitive waters are usually found at moderate to high elevation, in areas of high relief, with flashy hydrology and minimal contact between drainage waters and soils or geologic material that may contribute weathering products to solution. Sensitive streams are generally low order.

After passage of the Clean Air Act in 1970 and subsequent Amendments in 1977 and 1990, emissions and deposition of S were reduced and the concentrations of SO_4^{2-} in lake and streamwater in many areas in the eastern United States and Canada decreased (Dillon et al. 1987, Driscoll et al. 1989, Sisterson et al. 1990). Long-term monitoring data confirmed that much of the decrease in surface water SO_4^{2-} concentration was accompanied by rather small pH and ANC recoveries (Driscoll and van Dreaseon 1993, Kahl et al. 1993, Driscoll et al. 1995, Likens et al. 1996). The most significant response, on a quantitative basis, was decreased concentrations of calcium (Ca^{2+}) and other base cations. More recent data suggest increased surface water ANC during the 1990s in the Adirondack Mountains, Appalachian Plateau, and Upper Midwest, but not in the Ridge and Valley and Blue Ridge region of western Virginia (Stoddard et al. 2003).

As SO_4^{2-} concentrations in lakes and streams have declined, so too have the concentrations of Ca^{2+} and other base cations. There are several apparent reasons for this. First, the atmospheric deposition of base cations has decreased in recent decades (Hedin et al. 1994), likely due to a combination of air pollution controls, changing agricultural practices, and the paving of roads (the latter two affect generation of dust which is rich in base cations). Second, decreased movement of SO_4^{2-} through watershed soils causes reduced leaching of base cations

from soil surfaces. Third, soils in some sensitive areas have experienced prolonged base cation leaching to such an extent that soils may have been depleted of their base cation reserves. Such depletion greatly prolongs the acidification recovery time of watersheds and may adversely impact forest productivity (Kirchner and Lydersen 1995, Likens et al. 1996).

As aquatic effects researchers have revised their understanding of the quantitative importance of the various acidification processes, terrestrial effects researchers have also turned greater attention to the importance of the response of base cations to acidic deposition and the interactions between base cations (especially Ca^{2+} and Mg^{2+}) and Al. Likens et al. (1996) concluded that acidic deposition enhanced the release of base cations from forest soils at Hubbard Brook Experimental Forest (HBEF) in New Hampshire from the mid-1950s until the early 1970s, but that, as the labile pool of base cations in soil became depleted, the concentrations in streamwater decreased from 1970 through 1994 by about one-third. The marked decrease in base cation inputs and concomitant increase in net soil release of base cations at HBEF have likely depleted soil pools to the point where ecosystem recovery from decreased S deposition will be seriously delayed. Moreover, Likens et al. (1996) suggested that recently-observed declines in forest biomass accumulation at HBEF might be attributable to Ca^{2+} limitation or Al-toxicity, which can be expressed by the Ca^{2+} to Al^{3+} ratio in soil solution (Cronan and Grigal 1995).

Nitrate plays an important role in surface water acidification in some areas, especially with respect to episodic acidification during snowmelt or rainfall events (Wigington et al. 1993). This generally occurs in regions in which N deposition is sufficiently high that N supply exceeds the N demands of the forest vegetation and other biota, either chronically or episodically. Although most forests retain the majority of N inputs that they receive, some forested ecosystems leach significant amounts of NO_3^- to drainage water, and this occurs under a range of N deposition input levels. At some sites in the United States, relatively high levels of N deposition (10-30 kg N/ha/yr) have been shown to result in high N leaching losses (7-26 kg N/ha/yr). Good examples include watersheds in Great Smoky Mountains National Park in Tennessee (Johnson et al. 1991) and the San Bernardino Mountains in southern California (Fenn et al. 1996). At Fernow, West Virginia, high N loading (15-20 kg N/ha/yr) has caused moderate N leaching losses (~ 6 kg N/ha/yr; Gilliam et al. 1996, Peterjohn et al. 1996). Despite the observed variability in ecosystem response, a number of generalizations can be made (c.f., Sullivan 2000). Coniferous forests and alpine ecosystems seem to be more prone to N-saturation than deciduous forests

(Aber et al. 1995a, 1998; Williams et al. 1996). Young and successional forests, with their greater nutrient demand, are less prone to N-saturation than mature stands (Peet 1992). Climate, soil N pool size, and land use dramatically alter biological N-demand (Cole et al. 1992, van Miegroet et al. 1992, Feger 1992, Magill et al. 1996, Fenn et al. 1998). Because N is so critical to ecosystem primary productivity, other factors that influence plant and microbial growth (e.g., water availability, temperature) or the abundance or availability of other nutrients (e.g., Ca, Mg, P) will have significant effects on the ability of plants and microbes to utilize N and therefore the extent to which N will leach in drainage waters.

The relationship between measured wet deposition of N and streamwater output of NO_3^- was evaluated by Driscoll et al. (1989a) for sites in North America (mostly eastern areas), and augmented by Stoddard (1994). The resulting data showed a pattern of N leaching at wet-inputs greater than approximately 400 eq/ha (5.6 kg N/ha). This is about 22% higher than the 20-year average wet N deposition at Big Meadows in SHEN (4.6 kg N/ha), although wet N deposition at Big Meadows exceeded this amount in one year (6.6 kg N/ha in 1996) during the past decade and in three years over the period of record since 1981 (see Section V.C.3).

Stoddard (1994) presented a geographical analysis of patterns of watershed loss of N throughout the northeastern United States. He identified approximately 100 surface water sites in the region with sufficiently intensive data to determine their N status. Sites were coded according to their presumed stage of N retention, and sites ranged from Stage 0 through Stage 2. The geographic pattern in watershed N retention depicted by Stoddard (1994) followed the geographic pattern of N deposition. Sites in the Adirondack and Catskill Mountains in New York, where N deposition was about 11 to 13 kg/ha/yr, were typically identified as Stage 1 or Stage 2. Sites in Maine, where N deposition was about half as high, were nearly all Stage 0. Sites in New Hampshire and Vermont, which received intermediate levels of N deposition, were identified as primarily Stage 0, with some Stage 1 sites. Based on this analysis, a possible threshold of N deposition for transforming a northeastern site from the "natural" Stage 0 condition to Stage 1 would correspond to the deposition levels found throughout New Hampshire and Vermont, approximately 8 kg N/ha/yr. This agreed with Driscoll et al.'s (1989a) interpretation, which suggested N leaching at wet inputs above about 5.6 kg N/ha/yr, which would likely correspond to total N inputs near 10 kg N/ha/yr. This is likely the approximate level at which episodic aquatic effects of N deposition would become apparent in many

watersheds of the eastern United States. It is about 28% higher than the estimated total (wet + dry) N deposition at Big Meadows, which is about 7.8 kg N/ha/yr.

Comparable results were found in a survey of N outputs from 65 forested plots and catchments throughout Europe by Dise and Wright (1995). Below throughfall inputs (estimate of wet plus dry deposition) of about 10 kg N/ha/yr, there was very little N leaching at any of the study sites. At throughfall inputs greater than 25 kg N/ha/yr, the study catchments consistently leached high concentrations of inorganic N. At deposition values of 10 to 25 kg N/ha/yr, Dise and Wright (1995) observed a broad range of watershed responses. Nitrogen output was most highly correlated with input N ($r^2=0.69$), but also significantly correlated with input S, soil pH, percent slope, bedrock type, and latitude. A combination of input N (positive correlation) and soil pH (negative correlation) explained 87% of the variation in output N at 20 sites (Dise and Wright 1995). Thus, total N deposition of about 7.8 kg N/ha/yr at SHEN might be considered close to the adverse loading rates seen in other parts of North American and Europe.

2. Aquatic Fauna

Regional aquatic biodiversity in the southern Appalachian region is very high. Southern Appalachian streams contain a rich diversity of invertebrate, fish, and salamander species. Local species richness depends on thermal regime, water chemistry, patterns of discharge, plus substrate type and geomorphology (Wallace et al. 1992).

Acidification of waters in the Southern Appalachian Mountains region occurs against a backdrop of highly modified streams and rivers. About 98% of the free-flowing freshwater communities in the United States have been drastically altered, and only about 20% are of high enough quality to warrant Federal protection. To date, only about 1,600 km of streams and rivers have been given conservation status; only about 10% of these are east of the Mississippi River. Despite the highest national biological diversity in aquatic communities, southeastern streams have been afforded little protection. Geographic distributions of fish species have been greatly altered by humans, in part by inter-basin exchanges of water and especially by fish introductions, whether intentional or not. Logging and mining have been especially detrimental to fish, particularly brook trout (*Salvelinus fontinalis*). Many relatively healthy streams in second growth forests remain, but most large rivers have been dammed. Although biodiversity remains very high, it has likely been reduced in most aquatic communities in the Southeast. The

inability of flowing-water species to survive in the slow or still water habitats of lakes and ponds has caused much of this loss of diversity. Flowing-water species are specialized and very diverse, whereas lake communities are generalized and supplemented by introduced species. Human populations are growing in all southeastern states, and aquatic environments will be subject to increasing pressure. The streams in national parks and wilderness areas, including SHEN, may become increasingly important as biological refugia in the southern Appalachians.

Macroinvertebrates are defined as animals without backbones, which can be seen with the unaided eye, usually larger than 0.025 cm (0.01 inches) in at least one dimension. Aquatic benthic macroinvertebrates occur on the bottoms of streams or lakes, in or among substrates such as stones (gravel, cobble, etc.), plants, or wood. In lower order streams, the immature aquatic insects represent most of the macroinvertebrates, together with mollusks and crustaceans. The community contains many species of known sensitivity to stresses such as acidification or sedimentation. As with other groups, counts of taxa (such as families, genera or species) at impacted versus unimpacted sites are often lower due to loss of sensitive taxa, so lower species richness or absence of specific taxa is often taken to indicate impacts (SAMAB 1996).

Macroinvertebrate ecological roles in aquatic communities are diverse. Invertebrate species richness in the region is probably the greatest in North America, with many endemic species. Indeed, the regional invertebrate fauna includes many as yet undescribed species. The cool, high mountain streams in the region contain species that are usually only found further north. Many regional taxa have evolved rather elaborate morphological and behavioral adaptations for maintaining their positions in high-gradient streams with high current velocity (Wallace et al. 1992).

Benthic invertebrates play important roles in the breakdown of terrestrial and detrital material in streams and in nutrient regeneration. Many fish, waterfowl and other bird species rely on benthic invertebrates as their primary food source. Acidification, and consequent impacts of acidification on acid-sensitive benthic invertebrate species, can therefore affect structure, function, and trophic relations in stream ecosystems. Because benthic invertebrates are diverse, relatively easy to collect (though time-consuming to identify), and have short generation times, they are used as indicator species for a number of potential pollutants. This is true in the case of acidification and recovery from acidification, because communities typically contain species with known acid sensitivities (Baker et. al. 1990). While regional diversity of stream invertebrates is high, standing stocks are typically low in high-gradient, low-order streams

(Elwood et al. 1991). Production is also low (for example, 0.5-9.2 g dry weight /m²/yr), especially compared to very high production rates in regional rivers, which can be as high as 427.6 g dry mass/m²/year in the New River below Bluestone Dam in Virginia. Secondary production is a measure of the rate at which animal biomass is produced, regardless of its fate (e.g., loss to predators, natural mortality, emigration). Its units are biomass or energy per unit area, per unit time. Most studies of stream secondary production have been limited to a few taxa, and in high-gradient streams have focused on species with clearly discernable life cycles. Few studies have attempted to measure total macroinvertebrate production within a stream.

There are currently few data suggesting that acidification has produced significant changes in the invertebrate biota of southern Appalachian streams. This likely reflects the scarcity of long-term data, however, more than the absence of effects (Wallace et al. 1992).

The Ephemeroptera-Plecoptera-Trichoptera (EPT) index is one measure of stream macroinvertebrate community integrity. This is the total number of families in the three insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) present in a collection. These orders contain families of varying acid sensitivity so the index value (the number of families) is lower at acidified sites. SAMAB (1996) used the work of Mark Hudy in the George Washington and Jefferson National Forests (Virginia) to illustrate the effects of acidification on macroinvertebrates. EPT indices were calculated for 110 sites in George Washington and Jefferson National Forests based on collections made between 1992 and 1995. Among the 27 sites which had low EPT scores (eight or fewer families, EPT= 8 or less), about 60% had ANC values less than 100 µeq/L. The remaining stream sites with low EPT scores had other impacts. Sites with EPT scores of 14 or more had no conspicuous impacts.

Other common impacts might include sedimentation and pesticides. Excess fine sediments are detrimental to aquatic macroinvertebrates, as they are to fish, because they clog the interstices among bottom stones where macroinvertebrates take refuge. Excess fine sediments therefore may deprive organisms of food, as well as clogging the gravel nests where many species lay eggs. Aquatic insects are often sensitive to insecticides used in agriculture or silviculture. Low elevation stream sites in the southern Appalachians are often surrounded by agricultural land, often with no riparian forest buffer.

Known recent impacts other than acidification are absent in the case of the St. Marys River, located 30 km south of SHEN, which is unique in Virginia in having historical records of both water quality and benthic macroinvertebrates (SAMAB 1996). Indeed, it is the first clearly

documented case of biodiversity loss over time due to acidification in a southeastern U. S. stream (Kauffman et al. 1999). The St. Marys River is in the St. Marys Wilderness, George Washington and Jefferson National Forests in Augusta County, VA. This stream was sampled in the 1930s by E. W. Surber and in the 1970s-1990s by the Virginia Department of Game and Inland Fisheries. The stream has been acidified by atmospheric deposition, with pH declining from 6.8 in 1936 to 5.2 in 1988. Over the same time period, acid-sensitive insects declined or disappeared, with the number of taxa dropping from 31 to 18. The EPT index (based on genera in this case) declined from 17 to 10 (SAMAB 1996). Since 1988, the invertebrate species richness has continued to decline. The Shannon diversity index for benthic macroinvertebrates was 3.94 in 1936, 2.02 in 1988, and 1.8 in 1998; this index is usually between 3 and 4 for unpolluted streams. The observed declines in invertebrate taxa are consistent with those reported in the literature for streams with low pH. These taxa include *Epeorus*, *Ephemerella*, *Paraleptophebia* and *Stenonema*. Taxa known to be acid-tolerant (*Leuctra* and Chironomidae) have replaced them. This clearly reflects community degradation due to acidification, which may have occurred simultaneously in many similar streams in the southern Appalachians for which there are no historical records (Kaufmann et al. 1988).

The southern Appalachian area is widely regarded as one of the most diverse landscapes in the Temperate Zone (SAMAB 1996). Fish diversity is quite high. There are about 950 freshwater fish species in North America (Jenkins and Burkhead 1993), of which about 485 species can be found in the Southeast, about 210 species in Virginia (Jenkins and Burkhead 1993), and more than 30 species in SHEN. Regional habitat diversity and intraspecific genetic diversity are also regarded as high. Thus, the Southeast is a unique national biodiversity resource for fish. Unfortunately, the Southern Appalachian Assessment concluded that 70% of sampled stream locations showed moderate to severe fish community degradation, and that about 50% of the stream length in West Virginia and Virginia showed habitat impairment (SAMAB 1996).

Fish communities of high-gradient southern Appalachian streams may contain a variety of species, but are often dominated by trout, especially brook trout. Of the 15.1 million ha (37.4 million ac) in the southern Appalachian region (as defined by SAMAB 1996), 5.9 million ha (14.6 million ac [39%]) are in the range of native brook trout, with up to 53,000 km (33,000 mi) of potential native brook trout streams. This includes over 19,000 km (12,000 mi) of trout streams in Virginia (SAMAB 1996). There has been little regional ecological research on other species except in biogeographic and systematic studies, although Jenkins and Burkhead (1993)

provided much ecological information on the fish species of Virginia. There are, nevertheless, clear patterns in species distribution from headwaters to rivers, which can also be seen in community comparisons among reaches at different elevations; the clearest pattern is that species richness increases in a downstream direction. This is thought to result from the rather small number of upstream species, which must tolerate simultaneously highest current velocities and lowest pH values. Fish are absent from the highest headwaters, where they are replaced by salamanders. The highest-elevation fish species is usually brook trout, typically joined downstream by dace (e.g., blacknose dace, *Rhinichthys atratulus*), a sculpin (e.g. mottled sculpin, *Cottus bairdi*) and a darter (e.g. fantail darter, *Etheostoma flabellare*), and perhaps by introduced brown (*Salmo trutta*) or rainbow (*Oncorhynchus mykiss*) trout (Wallace et al. 1992). In the context of acidification, the introduced trout are both more acid-sensitive than brook trout, and will not be present in acidified waters.

Proceeding downstream, other dace, darters, chubs, shiners, suckers and others are often present. In larger downstream reaches, still regarded as high-gradient, the important gamefish smallmouth bass (*Salmoides dolomieu*) is most abundant in riffles over substrate which is about 40% clean gravel, boulder or bedrock, and at gradients of 0.8-4.8 m/km and depth at least 1.2 m (Wallace et al. 1992).

The fish of the southern Appalachians are all primarily insect predators. Trout, some dace, and some chubs are midwater and surface feeders, catching drifting aquatic invertebrates and terrestrial insects. Sculpins, darters, most chubs and minnows, and some dace feed primarily on benthic invertebrates, searching on and in the rock and gravel streambed, and some overturn rocks in their search. Because of limited primary production in such streams (due to shading across their entire width in summer), herbivores such as stonerollers occur only in somewhat larger streams with open canopy and lower gradient. Detritivore fish are uncommon in high-gradient streams in the region (Wallace et al. 1992).

Estimates of fish predation pressure on stream invertebrates suggest that pressure is substantial, but not more than invertebrate production. The fish community as a whole and brook trout in particular depend heavily on allochthonous (terrestrial) production, and terrestrial insects may make up 50% of trout diets. This terrestrial connection is direct in the case of fish feeding on terrestrial insects, and indirect in the case of stream invertebrate prey feeding on terrestrial detritus. Thus, effects on fish resources can be attributed in part to the alterations of water quality (acidification, sedimentation) and also to removal of terrestrial energy and food

additions through activities such as forest removal. Most small, high-gradient southern Appalachian streams, especially those that drain crystalline bedrock, have very low invertebrate production. A considerable portion of this production goes to predaceous invertebrates. In small, fishless, headwater streams, production of salamanders is similar to fish production in larger downstream reaches. Secondary production of carnivorous invertebrates is likely to be strongly influenced by local availability of food resources (Wallace et al. 1992).

In addition to stream chemistry, four factors are related to patterns of distribution and abundance of aquatic biota: temperature, gradient, stream order, and flow regime. Streams are divided into cold versus warm water; cold water streams have temperatures that rarely exceed 24-26° C for extended periods, and are characterized by the presence of trout and sculpin. Even at low temperatures, trout have relatively high metabolic rates, and are more active than most fish. They can use food resources more effectively than other fish at cold or cool temperatures (Moyle and Cech 2000).

Whereas temperature is of great importance in determining broad distribution patterns, gradient (number of meters drop per kilometer of stream) is often of greater local importance. This is because gradient has great influence on water velocity, substrate size, number and size of pools, and oxygen content. High-gradient streams may have little slow water, with bottoms of bedrock, boulders and cobbles, and few deep pools (Moyle and Cech 2000).

Stream gradient and temperature both vary with stream order, a classification of streams based on branching patterns. In most river systems, first order streams are the highest, smallest, coldest, highest-gradient streams, with fewest species of fish. As stream order increases, species richness also increases, usually as a consequence of increase in habitat diversity; turbidity, temperature, and stream size increase as well. As stream order increases, usually new species of fish are added at a higher rate than upstream fish are subtracted from the community (Moyle and Cech 2000). The trophic structure of the fish community also changes with stream order. In first order streams, the dominant fish (e.g. trout) usually feed on insects that drop into the water from the overhanging vegetation or on detritus. In higher-order streams, predators of aquatic insects are added, then piscivores. However, most stream fish are opportunistic feeders and may feed on a wide variety of foods (Moyle and Cech 2000).

The flow regime has an overriding influence on fish communities with respect to temperature, stream order, and gradient. Extreme fluctuations in flow regime (droughts to floods) can reduce or eliminate fish species or communities. However, extirpation by flood is

extremely rare, since adults, at least of most native species, appear to be able to find refuge. Floods are a common cause of missing year classes in populations, causing larvae or young to be washed downstream. Periods of low water may eliminate species through lowered oxygen levels, higher temperature, or aquatic vegetation growth, and the pre-drought community returns only gradually after typical conditions return (Moyle and Cech 2000).

There is a general pattern of increasing fish species richness and abundance from lower-order to higher-order streams, probably resulting in part from a greater variety of habitat types (including spawning and nursery areas) and food sources downstream. Perhaps because of the greater possibility of isolation, low-order streams are more likely to host species unique to each drainage (Adams and Hackney 1992).

Streams change continuously in physical and chemical characteristics from headwaters to river mouth. Changes include shift from primarily terrestrial to in-stream organic matter contributions, nutrient and water retention times, water volume and velocity, oxygen content, substrate size, gradient, and temperature. These shifts play important roles in determining the communities in different river sections. For example, localized differences in gradient have great effects on fish and invertebrate occurrences, and may result in markedly discontinuous distributions of individual species (Adams and Hackney 1992).

Physical complexity in streams affects biological processes chiefly through the availability of stable substrates that provide habitat and attachment surfaces. For example, in low-order streams, plankton are virtually non-existent, with consumers dependent on detrital materials or periphyton production on immobile substrates. In addition, rocks and woody debris provide the majority of physical complexity. In contrast, lower-order streams often exhibit high within-stream spatial variability in habitat conditions (Adams and Hackney 1992).

The physical characteristics related to basin morphology often affect biological processes in many ways, the most important of which relate to light and nutrients. In the continuum from headwater to river, headwater streams have the greatest interconnections with the terrestrial environment, and allochthonous (originating outside the stream system) energy sources typically dominate energy budgets; the majority of allochthonous material enters headwater streams as direct leaf litter, which is processed in place or downstream by macroinvertebrates. Autochthonous (in-stream) production is limited in these streams by dense shading by riparian vegetation, and nutrient limitation (Adams and Hackney 1992).

Wallace et al. (1992) reviewed rates of in-stream primary productivity in high-gradient streams in the Southeast and concluded that the most important controlling variables were stream order, season, degree of shading, and nutrients. It is clear that these variables can be intercorrelated, i.e., low-order streams are likely to be shaded across their entire width in forested catchments, and have lower nutrient content than downstream habitats, which are the beneficiaries of more weathering products. They found that unshaded reaches within high-gradient streams had three times higher primary productivity than did shaded reaches. Rates were 1.3 to 9 g C/m²/yr in several forested second- and third-order streams in Virginia and North Carolina including both soft (low Ca²⁺ and Mg²⁺) and hardwater (high Ca²⁺ and Mg²⁺) streams. Periphyton production in a hardwater (high base cations) Tennessee stream was found to be about 2.6 times higher than in a similar-sized softwater (low base cations) stream in North Carolina. While these differences in algal productivity are likely to be important, they are dwarfed by rates of periphyton production in the broad, sixth order New River in Virginia, where rates were up to 1059 g C/m²/yr in hardwater reaches, which were three to five times the rates in softwater reaches. Thus, whereas softwaters have lower in-stream production rates, both hardwater and softwater mountain streams have rather low rates of production compared with larger streams.

E. TERRESTRIAL INDICATORS

Bioindicators are those species for which pollutant sensitivity has been documented and for which data exist on their dose-response to pollutants and/or on symptomatology. In some cases, bioindicators detect exposure to a pollutant at a site where air quality monitoring data are not available. Ozone and SO₂ are the most extensively studied pollutants regarding impacts on vegetation. However, identifying symptoms of air pollutant injury is difficult, because visual symptoms are generally poorly documented, and only a small fraction of the thousands of native plant species have been screened for sensitivity to air pollutants.

Gaseous air pollutants can reduce photosynthesis, growth, and productivity of sensitive plant species (Treshow and Anderson 1989, Smith 1990, Runeckles and Chevone 1992, Chappelka and Chevone 1992), even at relatively low exposure levels (c.f., Reich and Amundson 1985). Some species are sensitive to episodes of high ambient exposure, while others are sensitive to lower chronic exposure over an extended period of time. Plants that are

physiologically stressed from air pollutants can have reduced vigor, which in turn can lead to greater susceptibility to additional stresses such as insect attack (Innes 1993, Pronos et al. 1999).

1. Plant Symptomatology

Pollutants can cause injury to various plant tissues including leaves, stems, and roots. Foliar injury is the most visible form of damage, although it often can be confused with other biophysical injuries, including abrasion, desiccation, insect herbivory, and fungal pathogens.

Ozone symptoms in conifer, hardwood, and herbaceous foliage that could be considered "typical" include chlorosis, stipple (change in color of epidermal cells, usually brown or purple, with no necrosis), and accelerated needle and leaf loss (Miller et al. 1983; Thompson et al. 1984a,b; Hogsett et al. 1989; Treshow and Anderson 1989; Stolte 1996; Brace et al. 1999). Common symptoms of O₃-induced foliar injury are summarized in Table III-1 (Krupa and Manning 1988, Krupa et al. 1998, Sullivan et al. 2001).

Ozone causes injury to highly-sensitive species of plants at concentrations as low as 60 ppbv (Treshow and Anderson 1989). Ozone enters plant leaves as a gas and dissolves in the presence of water. The resulting free radicals oxidize proteins of cell membranes, including those of the thylakoid membranes where photosynthesis takes place. Injury includes leaf discoloration, reduced photosynthetic rates, lowered sugar production, reduced growth, and possibly death (Sullivan et al. 2001).

Acute injury, such as visible injury on leaves, usually occurs after exposure to high concentrations of O₃. Moderate concentrations of O₃ (similar to ambient levels in much of the eastern United States) for several days to several months can cause chronic injury, accelerated aging, premature casting of foliage, or growth loss without other symptoms (Pell et al. 1994a,b; U.S. EPA 1996a,b). Accelerated aging may result in premature color change and loss of foliage, an effect of considerable importance at SHEN since park visitation levels are relatively higher during the fall color season. Growth reductions at ambient levels of O₃ are often difficult to measure, although a cumulative stress over multiple growing seasons may significantly reduce the growth and productivity of trees and understory vegetation (Reich and Amundson 1985, U.S. EPA 1996b).

There is considerable genetic variability, both within and among species, in response to O₃ (Miller et al. 1982). The most sensitive plants are injured by exposure to concentrations of 0.06 ppm or less for several days and can often be used as biological indicators of O₃ exposure (U.S.

Table III-1. Common symptoms of O ₃ -induced foliar injury (from Krupa and Manning 1988, Krupa et al. 1998, Sullivan et al. 2001).	
Acute Injury	Chronic Injury
<u>Conifers</u>	
<ul style="list-style-type: none"> • Banding; clear bands of chlorotic tissue develop on semi-mature needle tissue following O₃ episodes. • Tipburn; characterized by dying tips of young elongating needles. At first reddish-brown in color later turning brown, injury spreading 	<ul style="list-style-type: none"> • Flecking^a and mottling^b; flecking is the earliest symptom on the older needles of conifers. Mottling is generally associated with diffuse chlorotic areas interspersed with green tissue on first-year needles. • Premature senescence; early loss of needles.
<u>Broad-leaved plants (hardwood trees, herbaceous plants)</u>	
<ul style="list-style-type: none"> • Bleaching (unifacial or bifacial); small unpigmented necrotic spots or more general upper surface bleaching. Palisade cells and, where injury is more severe, upper epidermal cells collapse and become bleached. • Flecking; small necrotic areas due to death of palisade cells, metallic or brown, fading to tan, gray or white. • Stippling; small punctate spots where a few palisade cells are dead or injured, may be white, black, red, or red-purple. • Bifacial necrosis; when the entire tissue through the leaf is killed, bifacial dead areas develop, ranging in color from white to dark orange-red. While small veins are usually killed along with the other tissue, larger veins frequently survive 	<ul style="list-style-type: none"> • Pigmentation (bronzing); leaves turn red-brown to brown as phenolic pigments accumulate. • Chlorosis; may result from pigmentation or may occur alone as chlorophyll breakdown. • Premature senescence; early loss of leaves, flowers, or fruit.
^a Flecking is necrosis of tissue in small areas that may coalesce ^b Mottling is variation in chlorosis over the leaf surface, without necrosis	

EPA 1996b). Conversely, some plants are very tolerant of O₃ and are unaffected, even in severe exposures. Plants generally respond to lower O₃ concentrations than do people (U.S. EPA 1996b).

The timing of the O₃ exposure *vis-à-vis* the life cycle of the plant is important in determining response. For instance, in an annual plant before flowering, exposure to O₃ may reduce the growth of the root system; as the plant repairs injury in the leaves and maintains photosynthesis there is less carbon available to grow roots (U.S. EPA 1996b). In fact, root systems may be reduced in growth long before deleterious effects are manifested on above-ground portions of the plant. In perennial plants, the process is complex, as stored reserves are

usually available for growth and the effect of an exposure may not be manifested for several growing seasons (Hogsett et al. 1989, Andersen et al. 1991, Laurence et al. 1994). However, in the most sensitive species, a single season of exposure can be sufficient to reduce growth significantly (Wang et al. 1986, Woodbury et al. 1994).

Ozone may also affect the growth of plants indirectly through interactions with pests and pathogens (Laurence 1981), or by altering the symbiotic relations between plants and associated organisms (McCool 1988; Stroo et al. 1988; Andersen and Rygiewicz 1991, 1995; Rygiewicz and Andersen 1994). The resulting changes in nutrient availability or uptake may also result in altered plant growth, in this case mediated by O₃ exposure (Weinstein et al. 1991, Andersen and Scagel 1997, Weinstein and Yanai 1994).

Simulation modeling may be used to study the impact of O₃ on forest ecosystems, to establish the bounds of expected changes in growth, and to project shifts in ecosystem composition and structure (Laurence et al. 2001). While simulation modeling is unlikely to project exactly the response of any particular stand, the results, along with estimates of uncertainty in the projections, can be used to characterize potential changes and indicate the possible severity of effects (Woodbury et al. 1998).

Sulfur dioxide (SO₂) is a primary pollutant, given off during the combustion of fossil fuels, particularly coal, for the generation of electricity. The past three decades have seen tremendous advancements in control technologies and subsequent reductions in the emissions of this pollutant. For the most part, direct impacts of SO₂ on vegetation are a problem of the past in the United States. Indirect effects, through acidification of rain and deposition to forest ecosystems, while reduced, are still important. See Section VI.B.

Sulfur is an essential element for plant metabolism, and there are metabolic pathways for transformation of S. Plants generally take up S as SO₄²⁻ from the soil, not in a gaseous form. If SO₂ enters the leaf, it is converted to SO₄²⁻. During the transformation, sulfite, a compound highly toxic to plants, is produced. If the atmospheric concentrations of SO₂ are high (>about 0.5 ppm for 3 hours), toxic concentrations of sulfite may accumulate and cause injury before they are transformed to SO₄²⁻. If the exposure occurs over an extended period, SO₄²⁻ may also accumulate to toxic levels in the leaf.

Injury induced by SO₂ includes interveinal necrosis, dieback of leaf and needle tips, and necrotic spots (Thompson et al. 1980, 1984a,b; Treshow and Anderson 1989; Legge et al. 1998).

Unless a major S emissions source is located nearby, SO₂ rarely occurs in sufficiently high concentrations to cause direct damage to vegetation (Garner et al. 1989, Johnson et al. 1992).

There are few data or descriptions of the direct effects of NO_x on plant species, with the best description of symptomatology in Bytnerowicz et al. (1998). Concentrations of NO and NO₂ at locations remote from point sources are rarely high enough to cause visible injury symptoms. Experiments have been conducted to determine the relative sensitivity of plants to both nitrogen dioxide and nitric oxide, but the concentrations necessary to induce injury are far greater than those typically reported for ambient air quality, even in urban environments (NAS 1977). The adverse effects from long-term exposures can be reduced photosynthesis and growth in the absence of visible injury (Taylor 1968, Saxe 1994). At moderate levels, deposited N generally stimulates plant growth.

2. Physiological Effects

Considerable variation in the response of trees to air pollutants is caused by differences in the pollutant dose, phenological stage, age of leaves exposed, seed source, nutritional status of plants, and the integrated effects of multiple stresses (Bytnerowicz and Grulke 1992). There are several mechanisms for phytotoxicity, depending on the type of air pollutant. Physiological effects at the cellular and subcellular level can significantly affect plant function in the absence of visible symptoms of injury.

Gaseous air pollutants typically enter the plant through stomata, so the duration of stomatal opening greatly affects the pollutant dose assimilated. Once a pollutant enters a plant cell, there are many biochemical processes that are potentially affected (Wellburn 1988, Heath and Taylor 1997). Ozone increases the potential for the formation of harmful free radicals. Many of these free radicals are highly reactive and disrupt various metabolic processes through oxidation, substitution for other compounds, and toxicity. Disruption of photosynthetic processes (including damage to mesophyll cells and degradation of chlorophyll and chloroplasts [Grulke et al. 1996]) is one of the most deleterious effects of air pollutants, because it results in lower photosynthetic rates (Patterson and Rundel 1989, Grulke 1999), rapidly reduces plant vigor and productivity, and can affect internal resource allocation in the absence of visible symptoms (Matyssek and Innes 1999).

The degree of pollutant injury depends on the effective dose, which is a function of concentration, length of exposure, and stomatal aperture (Kozlowski and Constantinidou 1986).

Ozone injury of cells and tissues is essentially the same in woody and herbaceous plants (Bytnerowicz and Grulke 1992). Injury generally occurs first in the most photosynthetically active tissues, with disruption of chloroplasts in the palisade and mesophyll tissue. The loss of photosynthetic tissue results in visible chlorosis and necrosis.

Injury development in plants subjected to elevated concentrations of N compounds is not well understood, although it has been shown that acidic precipitation, particularly nitric acid vapor, can injure the cuticular layer and substomatal cavities. The contact time of acidic droplets or films on the leaf surface determines the degree of damage (Wellburn 1988). Acid-induced injury under experimental conditions typically requires exposure to very acidic solutions ($\text{pH} < 3$; Temple 1988, Turner et al. 1989), and injury in the field is rarely observed.

Photochemical reactions involving the absorption of sunlight and interactions with hydrocarbons and oxygen cause atmospheric NO_x to be consumed in the production of O_3 , with peroxyacyl nitrates (PANs) given off as secondary pollutants (Taylor et al. 1975). PANs have been shown to be highly phytotoxic (Mudd 1975). They can cause glazing or bronzing of the lower surface of leaves, indicating damage of the mesophyll cells around the stomatal cavity. Sensitive species have been shown to be injured by levels as low as 20 ppbv for two to four hour exposure periods.

The relationship of pollutant exposure to seasonal variation in physiological activity can have a significant influence on physiological effects and visible injury in plants (Grulke 1999). Gaseous uptake generally is higher during periods when soil water potential is high. The potential for pollutant uptake is higher during periods when soil moisture is high and plants are most metabolically active. Therefore, injury is more likely to be manifested by O_3 exposure during the spring when gaseous uptake is higher. This finding has implications for how O_3 exposure indices are quantitatively compared to O_3 injury measurements. Interactions with insects, pathogens, and other pollutants can accentuate the stress complex for plants exposed to a particular pollutant (Bytnerowicz and Grulke 1992, Sullivan et al. 2001).

3. Nitrogen Saturation and Forest Decline

Nitrogen is an essential nutrient for both aquatic and terrestrial organisms, and is a growth-limiting nutrient in many ecosystems. For each ecosystem, there is an input level of N which will increase ecosystem productivity without causing significant changes in species distribution

or abundance. Above that input level, adverse effects can occur in both aquatic and terrestrial ecosystem compartments (Gunderson 1992, Stoddard 1994, Aber et al. 1998).

The N cycle is extremely complex and controlled by many factors besides atmospheric emissions and deposition. Also, N inputs that may be beneficial to some species or ecosystems may be harmful to others. Nitrogen inputs to forested ecosystems are typically derived from the atmosphere, either through natural N-fixation or from atmospheric deposition. However, the pool of available N used by plants and microbes within the watershed can originate either from an external (atmospheric) source or an internal source, via decomposition and mineralization of stored organic N. The amount of inorganic N released through mineralization is affected by both the size and quality of the soil N pool (Aber et al. 1998), which in turn can be affected by disturbance and land use.

Increased atmospheric deposition of N does not necessarily cause adverse environmental impacts. In most areas, added N is taken up by terrestrial biota and the most significant effect seems to be an increase in forest productivity (Kauppi et al. 1992). However, under certain circumstances, atmospherically-deposited N can exceed the capacity of forest ecosystems to take up N. In some areas terrestrial ecosystems have become N-saturated¹ and high levels of N deposition have caused elevated levels of NO_3^- in drainage waters (Aber et al. 1989, 1991; Stoddard 1994). This enhanced leaching of NO_3^- causes depletion of Ca^{2+} and other base cations from forest soils and can cause acidification of soils and drainage waters in areas of base-poor soils (Reuss and Johnson 1986).

Increased levels of N also have the potential to increase foliar N content, alter N metabolism in plants (including winter hardiness), alter soil chemistry and N cycling, affect mycorrhizae, modify soil microbial dynamics, and alter plant growth (Takemoto 2000, Takemoto et al. 2001). Furthermore, litter decomposition rates are higher in areas with highest N deposition, presumably because of lower C:N ratios in the forest-floor litter (Fenn and Dunn 1989, Fenn et al. 1998). This is a significant change in an ecosystem process that has implications for biogeochemical cycling in general.

There is concern that excess N deposition could cause long-term terrestrial ecosystem damage that may be difficult to detect. In particular, both empirical and theoretical evidence

¹ The term N-saturation has been defined in a variety of ways, all reflecting a condition whereby the input of N (e.g., as NO_3^- , NH_4^+) to the ecosystem exceeds the requirements of terrestrial biota and a substantial fraction of the incoming N leaches out of the ecosystem as NO_3^- in groundwater and surface water.

suggest that lost species diversity may result from excess N inputs (Galloway et al. 1995). Increased N inputs may favor some species over others, thus changing successional dynamics. Acid (including N) deposition appears to make red spruce (*Picea rubens*) more susceptible to winter damage (NAPAP 1991, Johnson et al. 1992), at least in the northern part of its range. High SO_4^{2-} and NO_3^- levels can contribute to Al toxicity in red spruce populations. In addition, acidic cloudwater at high elevations can damage red spruce foliage, for example by reduced photosynthesis and reduced growth.

The end results of N-saturation can include forest decline, reduced forest growth, increased forest susceptibility to disease and insect infestation, eutrophication of estuaries and near-shore oceans, freshwater and soil acidification, loss of fish and other aquatic life, and changes in terrestrial and aquatic biodiversity. Fortunately, atmospheric N inputs to most forests are not high enough to cause such problems. Because of the severity of the potential effects, however, it is important that we understand the N cycle and the extent to which it is being perturbed by atmospheric emissions (Aber et al. 1998, Sullivan 2000).

N cycling is also regulated to a significant degree by climate, disturbance, and land management. Such factors are believed to have both short and relatively long-lasting (i.e., decadal to century) effects on the response of forest ecosystems to atmospheric N deposition (Aber et al. 1989, 1995b, 1998; Mitchell et al. 1996; Fenn et al. 1998). The extent to which the land was logged, burned, or used for agricultural production in the past can profoundly affect the N-status of the soils and therefore the extent to which N deposition will or will not cause environmental degradation.

It has recently been hypothesized that prior land use history, extending back 100 years or more, can have a major effect on forest response to N deposition (Aber and Driscoll 1997; Foster et al. 1997; Aber et al. 1997, 1998). The greater the previous extraction of N from the site by agricultural conversion, fire, logging, or other disturbance, the more N the forest will be able to absorb without becoming N-saturated. Aber et al. (1998) contended that previous land use is more important than either current or total accumulated N deposition as a controlling factor for N-saturation in the northeastern United States.

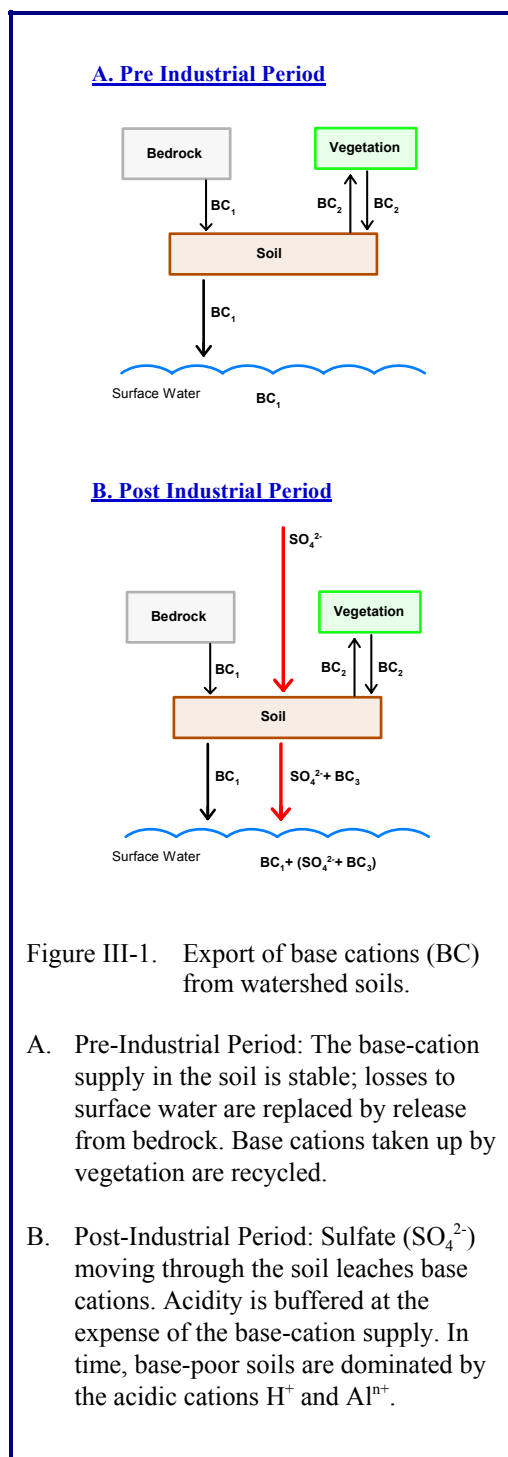
Forest stand age has also been found to be associated with N retention by the forest system. Watersheds having older trees seem more likely to leach N to a higher degree than forests having younger trees (c.f., Vitousek 1977, Elwood et al. 1991, Emmett et al. 1993).

4. Soil Acidification

Calcium and other base cations are major components of surface water acid-base chemistry, and are also important nutrients that are taken up through plant roots in dissolved form. Base cations are typically found in abundance in rocks and soils, but a large fraction of the base cation

stores are bound in mineral structures and are unavailable to plants. The pool of dissolved base cations resides in the soil as cations that are adsorbed to negatively-charged exchange sites. They can become desorbed in exchange for H^+ or Al^{n+} , and are thus termed exchangeable cations. The process of weathering gradually breaks down rocks and minerals, returning their stored base cations to the soil in dissolved form and thereby contributing to the pool of adsorbed base cations. Base cation reserves are gradually leached from the soils in drainage water, but are constantly being resupplied through weathering.

An increase in the concentration of SO_4^{2-} or other strong-acid anions in soil water will be balanced by an equivalent increase in the concentration of cations. Depending upon the availability of exchangeable base cations in the soil (primarily Ca^{2+} , Mg^{2+} , and K^+ ions), the cations associated with increasing concentrations of strong-acid anions in soil water can be either acidic or basic. The export of acidic cations (primarily H^+ and Al^{n+} ions) may contribute directly to loss of ANC, or soil water acidification. Although the export of base cations serves to reduce direct soil water and surface water acidification, it may also contribute to depletion of the base cation supply in the soil. Figure III-1 illustrates the process whereby S deposition leaches base cations from watershed soils. As the base-cation supply is reduced, the soil becomes more acidic and an



increasing proportion of the cation supply that is released from soils to soil water and surface water consists of H^+ and Al^{n+} ions (Reuss and Johnson 1986).

The supply of base cations to watershed soils can be external or internal. External sources include atmospheric deposition of base cations. Internal watershed sources are the main sources of base cations in most upland drainage waters of the eastern United States (Baker 1991). The primary internal sources of base cations in most watersheds are weathering and soil exchange. By comparison with exchange reactions, weathering occurs at relatively slow and constant rates (Turner et al. 1990, Munson and Gherini 1991). Thus, the main source of cations for acid neutralization in most watersheds is the accumulated supply of exchangeable base cations in the soil. However, the size of this supply, and thus the degree to which soil and surface water acidification occurs, is ultimately determined in large part by the weathering of base cations in watershed bedrock. As reflected in the low ANC and low base cation concentrations of streamwaters, most of the ridges in the central Appalachian Mountain region are underlain by base-poor bedrock (Webb et al. 1989, Church et al. 1992, Herlihy et al. 1993).

It is well known that elevated leaching of base cations by acidic deposition might deplete the soil of exchangeable bases faster than they are resupplied via weathering (Cowling and Dochinger 1980). However, scientific appreciation of the importance of this response has increased with the realization that watersheds are generally not exhibiting much ANC and pH recovery in response to recent decreases in S deposition (Lawrence and Huntington 1999, Driscoll et al. 2001a). In many areas this low level of recovery can be at least partially attributed to decreased base cation concentrations in surface water, which may indicate base cation depletion from soils.

There are few hardwood forests in which actual soil changes have been measured in association with acidic deposition (Raynal et al. 1992). Acidification and/or nutrient loss can be inferred, however, from studies of elemental budgets (c.f., Binkley and Richter 1987). There is a general tendency for some hardwood trees to accumulate Ca^{2+} , especially oak and hickory species, and this can cause Ca^{2+} depletion and soil acidification. Such effects can be exacerbated by both tree harvesting and acidic deposition (Johnson et al. 1988). Calcium limitation has not been shown to be significant in eastern hardwood forests to date, but several studies have suggested impending Ca^{2+} depletion with intensive harvesting (Johnson et al. 1988, Federer et al. 1989, Johnson and Todd 1990). The issue therefore merits continued monitoring.

Sulfur retention in watershed soils delays and/or reduces the potential for soil and surface water acidification in response to S deposition. This is particularly the case in the southeastern United States, where watersheds commonly retain more than 50% of deposited S (Rochelle and Church 1987, Turner et al. 1990), largely due to S adsorption on watershed soils. However, such S adsorption is capacity-limited, and as the adsorption capacity of soils becomes diminished, SO_4^{2-} concentration in soil waters and surface waters can increase, contributing to further acidification (Munson and Gherini 1991, Church et al. 1992).

Although soil acidification can occur in response to acidic deposition, soil acidification is also a natural process (Binkley 1989). The magnitude of the soil base saturation (an indicator of soil acid-base status, calculated as the portion of the cation exchange capacity that is comprised of base cations, as opposed to H^+ and Al^{n+}) is determined by the relative magnitude of base cation sources (weathering and atmospheric deposition) and sinks (leaching and plant uptake). If base saturation is low, or is observed to decline over time, this does not necessarily indicate an impact from acidic deposition.

F. VISIBILITY

The NPS monitors visibility conditions and supports studies to determine the causes of visibility degradation (haze) at many parks and wilderness areas nationwide. The purpose of this monitoring is to characterize current visibility conditions, identify the specific pollutants and their sources that contribute to visibility degradation, and document long-term trends to assess the effects of changes in pollutant emissions. The NPS cooperates and shares resources with other Federal land management agencies, states, and the U.S. EPA in the Interagency Monitoring of Protected Visual Environments (IMPROVE) Program.

1. Visibility Degradation

a. Sources of Visibility Degradation

Visibility can be altered by a variety of naturally occurring events, such as inclement weather, wildfire, or blowing dust, although visibility degradation as considered here excludes the effects of obscuration by weather. Visibility degradation results from the scattering and absorption of visible light by gases and particles in the atmosphere. If there are no suspended particles in the air, the clean-limit of natural visibility is determined by the amount of light scattered by “air molecules.” Scattering of visible light from “air molecules” falls within the

Rayleigh scattering regime, which means that the light is scattered evenly in both the forward and backward direction, with a preferential scattering of shorter wavelengths (blue light); this scattering is responsible for the blueness of the sky. The value of the Rayleigh scattering is a function of air density and elevation.

Particles in the atmosphere, derived from both natural and human sources, also scatter light. The amount of particle light scattering depends on the size and concentration of the particles, which in turn is affected by their physical and chemical properties. Fine particles (particles less than 2.5 μm in diameter) have a greater scattering efficiency, on a per mass basis, than coarse particles (particles between 2.5 and 10.0 μm in diameter). Particles with sizes near the wavelength of visible light (0.4 μm to 0.7 μm) scatter most efficiently.

Chemical composition of particles also plays a role in their relative scattering efficiencies. Categories generally used to differentiate between fine particle chemical species that scatter light are sulfates, nitrates, organics, and soil. Sulfate and nitrate aerosols are hygroscopic, that is, they absorb water at higher relative humidity conditions. Growth of these aerosols at higher relative humidity conditions can dramatically enhance their effect on scattering. Coarse particles are generally not speciated and are generally lumped together in terms of their scattering efficiency.

Some gases and particles can also absorb light. Nitrogen dioxide is the only major visible light-absorbing gas in the lower atmosphere. It usually does not occur in sufficient concentration in remote areas to make a major contribution to absorption. Elemental carbon (C; soot) is the dominant light-absorbing particle in the lower atmosphere.

b. Types of Visibility Degradation

Visibility degradation from atmospheric pollutants can generally be classified as 1) uniform haze, 2) layered haze, or 3) plume. A uniform haze generally obscures the appearance of landscape features or the sky, causes changes in the color or the contrast among landscape features, or causes features of a view to disappear altogether. Pollutants suspended in a section of the atmosphere can become visible, by the contrast or color difference between a layer or plume and a viewed background, such as a landscape feature or the sky. The view through a layered haze or plume will be altered in similar fashion to a uniform haze, but layered hazes and plumes are distinguished by a discernable boundary between the haze layer or plume and the adjacent cleaner atmosphere.

Pollutants mixed through the planetary boundary layer cause uniform hazes. This mixing typically occurs over several diurnal cycles and the resulting haze can extend for hundreds of kilometers. This condition is shown schematically in Figure III-2a. Pollutants observed far from an emissions source typically appear as elevated haze layers. Pollutants observed near the source of emissions often cause plume impacts (Figure III-2b). Plumes can occur under any wind and stability conditions, but are usually most severe with light wind speeds and stable thermal stratification. When plumes are emitted from elevated sources into a stable layer aloft, they can be transported for long distances with little vertical dispersion. Pollutants emitted into a stable layer at the earth's surface can form a ground-based layered haze (Figure III-2c). Layered hazes are usually associated with emissions that are more local in nature as opposed to pollutants that are transported over hundreds of kilometers. An observer positioned within a layered haze may perceive it as a uniform haze because layered haze boundaries can only be observed from an elevated position.

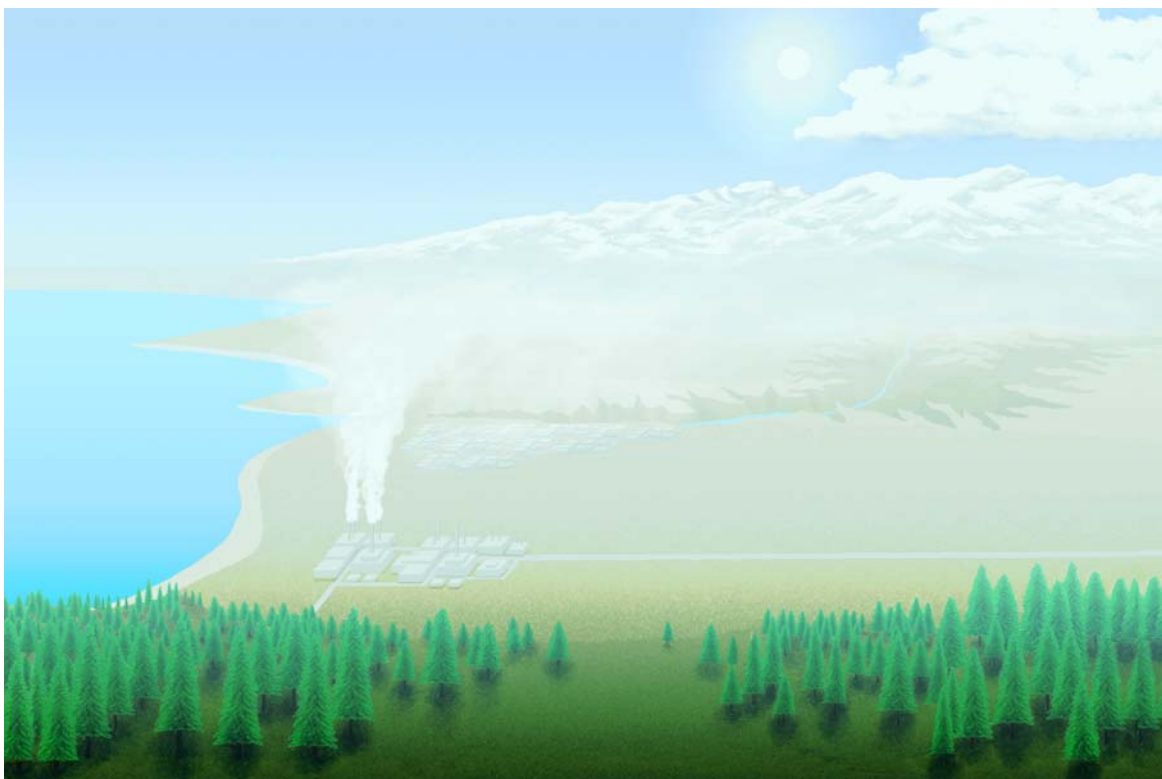


Figure III-2a. Uniform haze: one of three ways that air pollution can visually degrade a scenic vista.



Figure III-2b. Coherent plume: one of three ways that air pollution can visually degrade a scenic vista.



Figure III-2c. Layered haze: one of three ways that air pollution can visually degrade a scenic vista.

c. *Visibility Scales/Metrics*

To quantify visibility conditions and estimate the degree of degradation, several visibility scales/metrics can be employed. Depending on the application, metrics may include contrast and color difference index, light extinction coefficient (b_{ext}), visual range (VR), or degree of haziness.

For uniform haze circumstances, as depicted in Figure III-2a, where the pollutant loading becomes visible by contrast or color difference between the object luminance and the viewed background luminance, contrast or color difference metrics are generally used.

Light extinction coefficient, commonly called extinction, represents the attenuation of light per unit distance of travel in some medium, such as air, and is measured in inverse length (e.g., inverse kilometers [km^{-1}] or inverse mega meters [Mm^{-1}]). In the atmosphere, the b_{ext} comprises attenuation of light due to scattering and absorption of light by particles and gases in the air (aerosols).

Visual range is simply the greatest distance that a large, black object can be seen against a viewing background and is expressed as a distance (e.g., km or mi). Though easily understood and valuable for airport safety considerations, VR does not relate well to the perceived visual quality of a scene. A related metric is the Standard Visual Range (SVR). This metric is indexed to Rayleigh conditions at 1,800 m elevation rather than actual Rayleigh conditions at a specific location.

Visual range and the light extinction coefficient are inversely related (as VR decreases, b_{ext} increases) for certain assumed viewing conditions; a VR of 391 km signifies the best possible visibility (particle-free air, or Rayleigh conditions) and corresponds to a b_{ext} of 10 Mm^{-1} . These two metrics are usually used to quantify the level of degradation associated with uniform or ground-based layered hazes (Figures III.2a and c).

Neither VR nor b_{ext} is linear with respect to increases or decreases in perceived visual air quality. For example, a 15 km change in VR can result in a scene change either unnoticeably small or very apparent depending on the baseline visibility conditions. Therefore, another visibility metric, the haziness index (expressed as deciview [dv]), was defined to index a constant fractional change in the b_{ext} to visual perception (Pitchford and Malm 1994). The advantage of this characterization is that equal changes in dv are equally perceptible across different baseline conditions. Higher haziness index values signify poorer visibility. A zero dv corresponds to Rayleigh scattering (clean air), a VR of 391 km or a b_{ext} of 10 Mm^{-1} .

Of the three visibility metrics, the b_{ext} is the characterization most used by scientists concerned with the assessment of the causes of visibility degradation. Extinction can be directly calculated from light transmittance measurements (measured extinction) or derived from measured particle concentrations using linear relationships between the concentrations of particles and gases and their contribution to the extinction coefficient (calculated extinction). Understanding these relationships provides a method of estimating how visibility would change with changes in the concentrations of these atmospheric constituents. This methodology, known as “extinction budget analysis”, is important for assessing the visibility consequences of proposed pollutant emissions sources, or for determining the extent of pollution control required to meet a desired visibility condition. Extinction, both measured and calculated, is the primary visibility characterization provided in this report. The functional relationship among the three scales/metrics can be found in Appendix B.

2. Background on the Visibility Monitoring Program

The assessment of current visibility conditions in the eastern region for this report will focus on visibility degradation attributable to cumulative air pollutant emissions from numerous sources over a wide geographic area. Visibility degradation can, of course, be caused by local sources and specific meteorological conditions. However, robust data sets on individual coherent plumes or layered hazes in remote areas are generally not available. Such plumes or layered hazes must usually be evaluated on a case-by-case basis. Therefore, this assessment will focus on the conditions associated with regional or uniform haze.

In 1977, Congress established a national goal of no human-caused visibility impairment in Class I areas, and in 1999 promulgated a rule requiring states to develop and implement plans to make continuous progress toward that goal. The rule directs the states to use the b_{ext} coefficient as the scale/metric with which to demonstrate their progress. In particular, the states are to use the light extinction coefficient derived from measured particle concentrations and estimated specific extinction efficiencies for relevant particle species (calculated extinction, see discussion in Appendix B). The EPA has provided detailed guidance for assessing regional haze using this metric (U.S. EPA 2001a). Clearly, assessment of progress toward the national goal requires long-term particle monitoring on which to base estimates of natural and current visibility conditions. This long-term monitoring is available from the IMPROVE Program.

The IMPROVE Program is a cooperative monitoring effort that is governed by a steering committee composed of representatives from Federal and regional-state organizations. It was established in 1985 to aid the creation of Federal and state implementation plans for the protection of visibility in mandatory Class I areas, as stipulated in the 1977 amendments to the Clean Air Act. The objectives of IMPROVE are to:

1. establish current visibility and aerosol conditions in Class I areas,
2. identify chemical species and emission sources responsible for existing human-caused visibility impairment,
3. document long-term trends for assessing progress toward the national visibility goal, and
4. provide regional haze monitoring representing all mandatory Federal Class I areas included within the Regional Haze Regulations.

Ideally, a fully complemented IMPROVE monitoring station would include fine and coarse particle monitoring, optical monitoring, and view monitoring with photography. Each type of monitoring is briefly described below.

a. Particle Monitoring

Particle monitoring provides concentration measurements of atmospheric constituents that contribute to visibility impairment. Four independent IMPROVE sampling modules are used to automatically collect 24-hour samples of suspended particles every Wednesday and Saturday² by drawing in air and collecting suspended particles on filters. The filters are later analyzed to determine the chemical makeup of the suspended particles. Three of the four samplers (modules A, B, and C) collect fine particles with diameters $<2.5\ \mu\text{m}$. The fourth sampler (module D) collects particles with diameters up to $10\ \mu\text{m}$. Module A filters are analyzed to determine the gravimetric mass and elemental composition of the collected particles. Module B filters are analyzed specifically for SO_4^{2-} , NO_3^- , and chloride (Cl^-) ions (at SHEN, analysis for NH_4^+ has also been conducted since 1997). Module C filters are analyzed for organic material and light absorbing C. The gravimetric mass of coarse particles (2.5 to $10.0\ \mu\text{m}$) is determined by subtracting the Module A gravimetric mass from the Module D gravimetric mass.

² IMPROVE sampling schedules changed to a 1-in-3 day (EPA) schedule in 2000.

b. Optical Monitoring

Optical monitoring provides a direct quantitative measure of light extinction to represent visibility conditions. Water vapor in combination with suspended particles can affect visibility, so optical stations also record temperature and relative humidity. Optical monitoring uses ambient, long-path transmissometers or ambient nephelometers to collect hourly-averaged data. Transmissometers measure the amount of light transmitted through the atmosphere over a known distance (between 0.5 and 10.0 km) between a light source of known intensity (transmitter) and a light measurement device (receiver). The transmission measurements are electronically converted to hourly averaged light extinction (b_{ext} , scattering plus absorption). Ambient nephelometers draw air into a chamber and measure the scattering component of light extinction. Data from both of these instruments are recovered at a central location for storage and analyses. Optical measurements of extinction and scattering include meteorological events such as cloud cover and rain, but the data are "filtered" by flagging as invalid data points with high relative humidity (RH>90%). This filtering process is assumed to remove the largest effects of weather from the data set. Optical data also fulfill an important need by providing concurrent, independent measurement of extinction to compare with extinction coefficient calculations made using particle data.

c. View Monitoring

View monitoring is accomplished with automated 35 mm or digital camera systems. Cameras typically take three photographs a day (9:00, 12:00, and 15:00) of selected scenes. The resulting slides are used to facilitate data interpretation, and form a photographic record of characteristic visibility conditions. Based on April 1995 recommendations of the IMPROVE Steering Committee, view monitoring has been discontinued at all NPS Class I areas that have a five-year (or greater) photographic monitoring record. View monitoring has not been conducted at any of the NPS visibility monitoring sites in the Appalachian Region since October 1999.